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PERSONAL EXPOSURE TO FINE PARTICLES AND BENZO[A]PYRENE. RELATION WITH INDOOR AND OUTDOOR CONCENTRATIONS OF THESE POLLUTANTS IN KRAKÓW

WIESŁAW JĘDRYCHOWSKI[1], AGNIESZKA PAC[1], HYUNOK CHOI[2], RYSZARD JACEK[1], ELŻBIETA SOCHACKA-TATARA[1], THOMAS S. DUMYAHN[3], JOHN D. SPENGLER[3], DAVID E. CAMANN[4] and FREDERICA P. PERERA[2]

[1]Chair of Epidemiology and Preventive Medicine,
Jagiellonian University Medical College, Kraków, Poland
[2]Columbia Center for Children's Environmental Health,
Mailman School of Public Health, Columbia University, New York, USA
[3]Department of Environmental Health,
School of Public Health, Harvard University, Boston, MA, USA
[4]Department of Analytical and Environmental Chemistry,
Southwest Research Institute, San Antonio, Texas, USA

Abstract
Objectives: This study assessed personal exposure of pregnant women to fine particles ($PM_{2.5}$) and benzo[a]pyrene (B[a]P) and the relationship between pollutant concentrations in ambient and indoor air. **Materials and Methods:** In a group of 78 pregnant women, simultaneous 48 h measurements of personal, indoor, and outdoor exposure to $PM_{2.5}$ and B[a]P were carried out in the second trimester of pregnancy. The results show that participants were exposed to varying concentrations of $PM_{2.5}$ and B[a]P, with higher exposure in the winter season. Overall, the mean personal $PM_{2.5}$ level was 30.4 μg/m^3 and B[a]P 2.1 ng/m^3. The winter/summer ratios for mean personal exposures were 1.4 (35.6 μg/m^3 *vs*. 25.8 μg/m^3) and 5.4 (4.9 ng/m^3 *vs.* 0.9 ng/m^3), respectively. As for indoor levels, the winter/summer ratios were 1.4 (33.2 μg/m^3 *vs*. 24.4 μg/m^3) for $PM_{2.5}$ and 5.4 (4.3 ng/m^3 *vs*. 0.8 ng/m^3) for B[a]P, and for outdoor concentrations, the respective values were 1.5 (40.3 μg/m^3 *vs*. 26.4 μg/m^3), and 6.8 (6.1 ng/m^3 *vs.* 0.9 ng/m^3). A stronger correlation was found between personal $PM_{2.5}$ exposure and the pollutant concentration indoors (r = 0.89; 95% CI: 0.83–0.93) than outdoors (r = 0.75; 95% CI: 0.64–0.83). The correlations between personal B[a]P exposure and its indoor or outdoor levels were similar (0.95–0.96) and significant. The markedly higher exposure to B[a]P in Kraków in winter than in summer can be explained by the massive use of coal for heating in the cold season. **Conclusion:** We conclude that although ambient $PM_{2.5}$ measurements provide an adequate indicator of outdoor air quality for use in epidemiologic studies, they may not be adequate for studies on relationship between non-ambient pollution and health effects. Since only about 20% of variability in personal B[a]P exposure could be explained by personal $PM_{2.5}$ level, the extrapolation of personal exposure to B[a]P from personal $PM_{2.5}$ data may be greatly underestimated.

Key words:
Indoor, Outdoor urban air, Personal exposure, Fine particles, Benzo[a]pyrene

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Address reprint requests to W. Jędrychowski, MD, PhD, Chair of Epidemiology and Preventive Medicine, Jagiellonian University Medical College, Kopernika 7A, 31-034 Kraków, Poland (e-mail: myjedryc@cyf-kr.edu.pl).

INTRODUCTION

Monitoring data on the levels of pollutants in ambient air, such as annual or daily averages of total suspended particulates (TSP) or SO_2, have been widely used to identify and predict adverse health effects at the population level [1–13]. Numerous epidemiologic studies have found an association between health outcomes and air pollution level based on stationary ambient air monitoring. However, there is a concern that the ambient measurements do not reflect an important gradient of personal exposure across individuals and population groups. Therefore, environmental surveys currently focus more on the chemical composition and the size of particulates and consider the contribution of indoor sources, ventilation or daily activity in the assessment of personal burden resulting from exposure to specific air pollutants [14–16].

Fine particles ($PM_{2.5}$), with an aerodynamic diameter of less than 2.5 μm, are more likely than the coarse large particles to cause adverse health effects since they are deposited in the peripheral lung. Moreover, fine particles produced mostly by burning fossil fuels, may contain such toxic substances as lead, manganese, sulfates, and organic compounds. Of the airborne organic compounds, the polycyclic aromatic hydrocarbons (PAHs) have received most attention. Benzo[a]pyrene (B[a]P) and other PAHs, a complex group of chemicals generated during incomplete combustion of organic material, may pose serious health risks. They are found not only in automobile exhaust but also in cigarette smoke, industrial effluents, and in some charcoal-broiled foods [2,3,10,13].

The main purpose of the exposure assessment part of the current epidemiological study was to compare personal exposure to fine particles ($PM_{2.5}$) and B[a]P in Kraków inhabitants. However, an additional goal of the analysis was to assess the relation between the total personal exposure to these pollutants and their ambient and indoor levels in different seasons of the year and residential areas of the city.

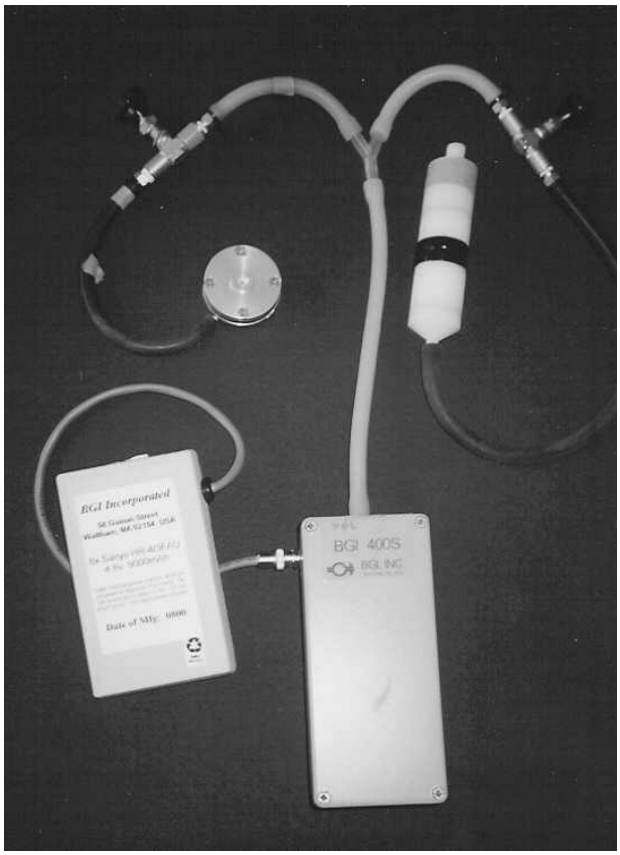
MATERIAL AND METHODS

This study uses data from the Kraków birth cohort of children as a part of a collaborative study with Columbia University, New York. A detailed description of the study has been published elsewhere [17]. Briefly, between November 2000 and November 2003, a total of 407 women were recruited from ambulatory prenatal clinics, and healthy pregnant women in the first and second trimester of pregnancy were eligible for the study. The enrollment criteria included only non-smoking women aged 18–35 years, with singleton pregnancies, with no illicit drug use and HIV infection, free from chronic diseases such as diabetes or hypertension, and residing in Kraków for at least one year prior to pregnancy. The recruited women were interviewed and received information with a description of the study and requirements for participation. Upon enrollment, a detailed questionnaire was administered to each subject to solicit information on demographic data, household characteristics, medical and reproductive history, occupational hazards, and smoking habits of other family/household members. In all women, a series of 48 h measurements were carried out in the second trimester of pregnancy. In the group of 85 pregnant women recruited for the study, indoor and outdoor measurements were collected at the time of their personal exposure to $PM_{2.5}$ and B[a]P. The study sample under statistical analysis consisted of 78 women who had personal, indoor, and outdoor measurements completed during the second trimester of pregnancy, and reliable results of these measurements were available.

Personal measurements were collected by placing the Personal Exposure Monitor Sampler (PEMS, Harvard School of Public Health) with teflon filter, 37 mm in diameter, in the backpack which the woman was instructed to wear during the day and place by her bed during the night (Fig. 1). Indoor air was collected in the room of the subject's home/apartment where there was the most activity. The sampling equipment placed outside the window, usually on a balcony, about one meter from the wall of the home, collected outdoor measurements. We used the same type of samplers to ensure comparability between all the three measurements. The sampling pumps (BGI, Waltham, MA, USA) were powered by re-chargeable battery of 32-h capacity. The pump with a split flow inlet was used for simultaneous collection of particles with an aerodynamic diameter ≤ 2.5 μm and vapors and aerosols on a polyurethane foam

(PUF) backup plug, at the flow rate of 2 liters per minute (Fig. 2). Flow rates were determined using DC-Lite Flow Meter (BIOS International Corporation) which was calibrated using Gilibrator™2 (Gilian®, SENSIDYNE Inc.) standards. Inlets of each sampler were placed outside the backpack close to the breathing zone. On the second day of monitoring, the air monitoring staff visited the subject's home to check that the monitor had been running continuously and that there were no operating failures, to change the battery-pack and administer related questionnaires.

Fig. 1. One of the participants of the environmental survey wearing the backpack with personal monitoring samplers for collection of $PM_{2.5}$ and B[a]P.



PEMS — personal exposure monitor sampler.
PUF — polyurethane foam.

Fig. 2. Sampling instruments placed in the backpack of the recruited subjects.

A staff member returned again to the woman's home on the third day to collect the equipment. Immediately after the samples had been collected, they were inventoried and kept at a low temperature at the University laboratory.

Once per month, air samples were shipped on dry ice to Harvard University (PEMS filters) or Southwest Research Institute (PUF filters) via the Center for Children's Environmental Health, Columbia University. At Harvard School of Public Health, the samples were conditioned for 48 h in a temperature- and humidity-controlled room (18–24°C, 40±5% relative humidity). Then they were weighted using microbalance (Mettler Model MT5) with detection limit of 6 μg. All the filters were weighted twice and the average weight was used as a filter weight. When the difference in the duplicate filters weight exceeded 5 μg, the filter was weighted again and the average of the two closest weights was used.

To ensure the quality of air monitoring and data analysis of $PM_{2.5}$ pollution, the samples were collected according to the standardized field study protocol. To avoid potential contamination resulting from improper handling, transport or storage, we included an analysis of blank and spiked samples with field samples. Each sample was coded as to the accuracy of flow rate, duration, and completeness of documentation. $PM_{2.5}$ concentration was calculated according to the formula: $PM_{2.5}$ concentration [μg/m^3] = 1000 × mass difference [μg]/(mean flow[dm^3/min] × time[min]). Mean flow was calculated as the average of mean flows during the first and second day of the measurement.

B[a]P was collected by the polyurethane cylinder (PUF sample cartridge). The PUF sample cartridge was preceded by an impactor inlet with a 2.5 μm cut at 2 LPM. Vapors and particles of ≤ 2.5 μm in diameter pass through the impactor inlet and collect on a precleaned quartz microfiber filter (Palliflex Tissuquartz 2500 QAS, 25 mm in diameter) and a precleaned polyurethane cylinder. After the sampling had been completed, the field samplers were frozen and shipped on dry ice to South-West Research Institute in Texas.

Laboratory analysis of B[a]P air monitoring samples

The samples were stored at –12°C until the extraction date. The quartz microfiber filter and PUF plug were removed, spiked with p-terphenyl-d_{14} (as an extraction surrogate), and soxhlet-extracted (Corning, Corning N.Y.) with 6% diethyl ether in hexanes for 16 h. Each extract was concentrated to a final volume of 1.0 ml and frozen at –12°C. The air extracts were analyzed by GC/MS for benzo(*a*)pyrene [18]. GC/MS analysis was performed using Agilent 6890 gas chromatograph/5973 mass spectrometer (Agilent Technologies, Palo Alto, CA) in selected ion monitoring mode to achieve low-level detection. Extracts were injected into a 0.25 mm inner diameter capillary analytical GC column (a 30 m DB-5.625 or a 60 m DB-5MS (Agilent Technologies, Palo Alto, CA)). As laboratory quality control, matrix blank, solvent blanks, and matrix spike samples were extracted and analyzed with each batch of samples. The recovery of the extraction surrogate, p-terphenyl-d_{14} was consistently above 60%. The 48 matrix spikes showed that B[a]P was efficiently extracted with recovery means above 91% and recovery standard deviations from 18% to 31%. The reported air concentrations were not adjusted for spike recoveries. Two laboratory analysts analyzed all samples using the same technique. The reliability of the analyses over the study period was also examined. Measurement agreement of the collocated duplicate samples was higher than 90%.

Statistical data analysis

Because of their skewness, the exposure data were log-transformed to normalize their distribution. Geometric means and geometric standard deviations were calculated to examine the $PM_{2.5}$ and B[a]P concentrations. We used t-test to examine differences in the level of air pollution between the heating and non-heating seasons and between two residential areas (old city center and outer city area). Measurements were divided into those for which the onset of measurements was between October 16th and March 15th (heating season) and those between March 16th and October 15th (non-heating season) because the municipal heating system of Kraków is usually on from Oct 16th to March 15th. To estimate the relation between personal exposure of the subjects and outdoor and indoor levels of pollutants, stepwise multiple regression models were used. In the statistical analysis of data, BMDP New System 2.0 (Statistical Solutions Ltd., Cork, Ireland) was used.

RESULTS

The characteristics of the study population, together with the distribution of potential environmental and home sources of air pollution are shown in Table 1. The subsample of subjects who were included in the air pollution study were young (mean age 27.6 years), more than 60% of them had a university/bachelor degree, and none was exposed to dusty occupational settings. About 50% of measurements were carried out in the heating season (winter months). Most of the subjects lived in the outer city area (82%) and 22% declared that they were exposed to environmental tobacco smoke (ETS) (based on the presence of 1 or more smokers in the household). The distribution

Table 1. Characteristics of study population

Variables	Study sample N = 78	Total sample N = 407
	N (%)	N (%)
Age*	27.6 (3.92)	27.6 (3.63)
Season of measurement		
Heating season	38 (48.7%)	175 (43.0%)
Non-heating season	40 (51.3%)	232 (57.0%)
Place of residence		
City center	14 (17.9%)	77 (18.9%)
Outer city area	64 (82.1%)	330 (81.8%)
Education		
University/bachelor	52 (66.7%)	266 (65.4%)
Lower education	26 (33.3%)	141 (34.6%)
Traffic density outside the window		
Window does not face the street	45 (57.7%)	277 (68.1%)
Small traffic density	13 (16.7%)	93 (22.8%)
Medium traffic density	14 (17.9%)	60 (14.7%)
High traffic density	6 (7.7%)	27 (6.6%)
Passive smoking		
Yes	17 (21.8%)	121 (29.7%)
No	61 (78.2%)	286 (70.3%)
Time spent outside (hours)*	2.7 (2.15)	3.3 (3.08)
Time spent in the transit (hours)*	1.8 (1.48)	2.6 (2.56)

* Mean (SD).

of all variables in the examined subsample was similar to the total sample.

Figures 3 and 4 present the cumulative distributions of personal, indoor, and outdoor 48-h concentrations of $PM_{2.5}$ and B[a]P for the subjects. On average, only about 9% of subjects were exposed to $PM_{2.5}$ level > 65 μg/m^3, which is

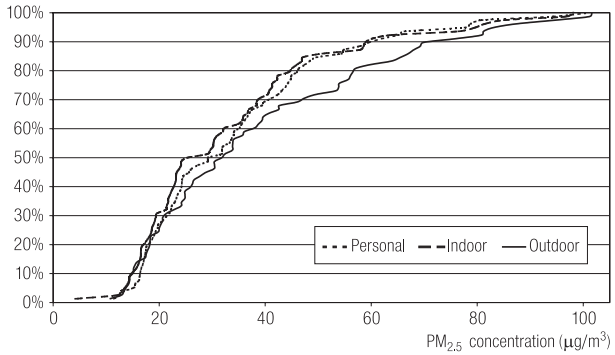


Fig. 3. Cumulative frequency of $PM_{2.5}$ concentration by type of measurement.

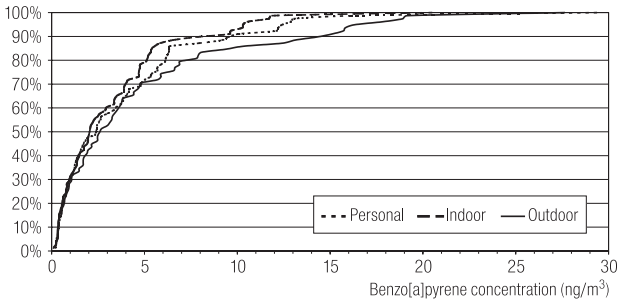


Fig. 4. Cumulative frequency of benzo[a]pyrene concentrations by type of measurement.

the EPA daily standard for $PM_{2.5}$ [18], while about 70% of women were exposed to levels of B[a]P >1 ng/ m^3.

The comparison of geometric mean concentrations of personal $PM_{2.5}$ and B[a]P exposure with those measured indoors and outdoors is presented in Table 2. It shows that the mean personal $PM_{2.5}$ exposure was 30.4 μg/m^3 (SD: 19.9) and was somewhat lower (by 7%) than its outdoor level (32.8 μg/m^3, SD: 24.1) and higher (by 7 %) than $PM_{2.5}$ indoors (28.5 μg/m^3, SD: 20.9).

We found a significant correlation between personal $PM_{2.5}$ exposure and its levels indoors (r = 0.89; 95% CI: 0.83–0.93) and outdoors (r = 0.75; 95% CI: 0.64–0.83). The difference between the two correlation coefficients was statistically significant (z = 2.749, p = 0.006). The stepwise multivariate regression of personal $PM_{2.5}$ exposure on independent variables (indoor and outdoor $PM_{2.5}$ levels, ETS) showed that the most important factor affecting variability of personal $PM_{2.5}$ exposure was its indoor component (r^2 = 0.57), followed by outdoor $PM_{2.5}$ level (r^2 = 0.26) and ETS (r^2 = 0.20).

Table 2. Geometric mean concentrations of $PM_{2.5}$ [μg/m^3] and benzo(a)pyrene [ng/m^3] at personal and ambient level (N = 78)

Type of measurement	Benzo[a]pyrene	$PM_{2.5}$
Personal	2.1 (5.0)	30.4 (19.9)
Indoor	1.9 (3.9)	28.5 (20.9)
Outdoor	2.4 (6.2)	32.8 (24.1)
Personal/indoor ratio	1.1	1.1
Personal/outdoor ratio	0.9	0.9
Indoor/outdoor ratio	0.8	0.9

In bracket SD.

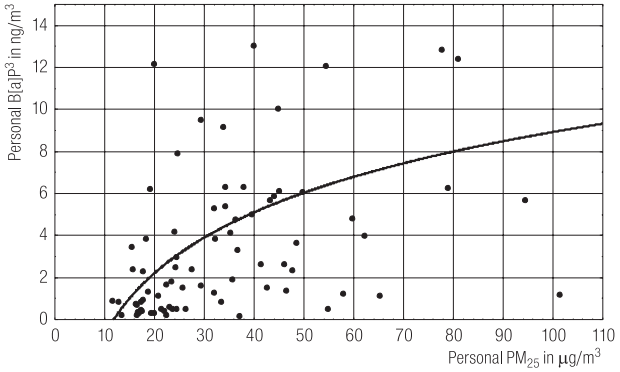


Fig. 5. Scatter plot and logarithmic fitting of personal $PM_{2.5}$ and B[a]P concentrations.

The average personal exposure to B[a]P was also somewhat lower (by 12%) than that observed outdoors (2.1 ng/m^3 *vs.* 2.4 ng/m^3) and higher (by 11%) than B[a]P indoors (2.1 ng/m^3 *vs.* 1.9 ng/m^3). The correlation between personal B[a]P level and the pollutant concentrations indoors or outdoors were similar (r = 0.91–0.93). Hence, up to 87% of variability in personal exposure to B[a]P could be explained by its ambient levels. The stepwise multivariate regression of personal B[a]P exposure on indoor and outdoor B[a]P levels showed that the most important factor influencing variability of personal B[a]P exposure was its outdoor component (r^2 = 0.51), followed by indoor B[a]P level (r^2 = 0.43). The correlation between personal B[a]P and personal $PM_{2.5}$ was 0.43 (Fig. 5).

There was a clear increase in $PM_{2.5}$ and B[a]P concentrations in the heating season (Table 3). The winter/summer ratio for personal $PM_{2.5}$ exposure was 1.4 (35.6 μg/m^3 *vs.* 25.8 μg/m^3), but the respective ratio for B[a]P exposure was 5.7 (4.9 ng/m^3 *vs*. 0.9 ng/m^3). Similarly, the winter/summer ratio for indoor $PM_{2.5}$ was 1.4 (33.2 μg/m^3 *vs.* 24.4 μg/m^3) and the ratio for B[a]P amounted to 5.4 (4.3 ng/m^3 *vs.* 0.8 ng/m^3). The comparison of the outdoor

Table 3. Geometric means (SD) of personal and ambient exposure to air pollutants in different seasons of the year

Measurement season	Type of measurement	Benzo[a]pyrene	$PM_{2.5}$
Summer N = 38	Personal	0.9 (1.5)	25.8 (18.7)
	Indoor	0.8 (1.2)	24.4 (20.5)
	Outdoor	0.9 (1.5)	26.4 (19.2)
	Personal/indoor ratio	1.1	1.1
	Personal/outdoor ratio	1.0	1.0
	Indoor/outdoor ratio	0.9	0.9
Winter N = 40	Personal	4.9 (5.6)	35.6 (23.9)
	Indoor	4.3 (4.1)	33.2 (24.0)
	Outdoor	6.1 (6.6)	40.3 (32.2)
	Personal/indoor ratio	1.2	1.1
	Personal/outdoor ratio	0.8	0.9
	Indoor/outdoor ratio	0.7	0.8

Table 4. Geometric means (SD) of personal and ambient exposure to air pollutants by the residential area of the subjects

Residential area	Type of measurement	Benzo[a]pyrene	$PM_{2.5}$
City Center N = 14	Personal	1.9 (5.4)	32.2 (26.7)
	Indoor	1.8 (4.4)	30.8 (21.1)
	Outdoor	2.3 (7.3)	35.7 (30.3)
Outer city area N = 64	Personal	2.2 (5.1)	30.0 (18.2)
	Indoor	1.9 (3.8)	28.1 (21.0)
	Outdoor	2.4 (6.0)	32.1 (22.7)

Table 5. Geometric means (SD) of personal and ambient exposure to air pollutants by the residential area and season of the year

Residential area	Measurement season	Type of measurement	Benzo[a]pyrene	$PM_{2.5}$
City Center	Summer N = 6	Personal	0.6 (2.6)	21.7 (8.0)
		Indoor	0.6 (2.1)	22.3 (7.2)
		Outdoor	0.6 (2.4)	24.8 (10.5)
	Winter N = 8	Personal	4.8 (5.2)	43.3 (29.9)
		Indoor	4.3 (4.3)	39.3 (23.8)
		Outdoor	6.1 (7.4)	46.8 (34.4)
Outer city area	Summer N = 32	Personal	0.9 (1.3)	26.6 (19.9)
		Indoor	0.9 (1.1)	24.8 (22.2)
		Outdoor	1.0 (1.4)	26.7 (20.5)
	Winter N = 32	Personal	5.0 (5.8)	33.9 (16.1)
		Indoor	4.3 (4.1)	31.8 (19.6)
		Outdoor	6.1 (6.5)	38.8 (23.0)

concentrations of air pollutants showed that the winter/summer ratios were 1.5 for $PM_{2.5}$ (40.3 μg/m^3 *vs.* 26.4 μg/m^3), and 6.8 for the B[a]P (6.1 ng/m^3 *vs.* 0.9 ng/m^3).

Table 4 shows that in the city center, outdoor $PM_{2.5}$ concentration was slightly higher than in the outer city area (35.7 μg/m^3 *vs.* 32.1 μg/m^3) but the difference was insignificant ($t = 0.840$; $p < 0.402$). Personal exposure to $PM_{2.5}$ and $PM_{2.5}$ indoors were also higher in the city center, but again the differences were not significant. These differences were slightly larger in the winter months (Table 5). The observed differences in B[a]P exposure across the city areas were very small and insignificant as well.

DISCUSSION

The study documented that the geometric mean of personal $PM_{2.5}$ exposure in the subsample of pregnant women from the Kraków inner city was 30.4 μg/m^3. The average observed concentration of personal $PM_{2.5}$ measurements compared with EPA daily standards (> 65 μg/m^3) showed that about 9% of subjects had personal $PM_{2.5}$ exposure > 65 μg/m^3. Mean personal B[a]P level was 2.1 ng/m^3 (±5.0) and about 70% of persons examined were exposed to the levels above 1 ng/m^3 (WHO standard). Up to now, the B[a]P standards have been subject to intensive debate. For example, the U.K. government's Expert Panel on Air Quality Standards (EPAQS) has recommended the U.K. standard for B[a]P of 0.25 ng/m^3 [19]. Although similar recommendation does not exist for Poland, the Swedish guideline value for B[a]P is 0.1 ng/m^3 [20].

It is worth noting that while the $PM_{2.5}$ concentrations during winter time increased by about 40% compared to summer months, there was about a > 5-fold increase in B[a]P. This discrepancy may result from the fact that a major contributor to B[a]P is coal use which highly increases during the winter heating season.

We observed that indoor $PM_{2.5}$ explained a greater portion of personal exposure variance than outdoor $PM_{2.5}$ did. The reason may be the generation of $PM_{2.5}$ by indoor sources such as ETS, oven cooking or frying, cleaning, etc. and the fact that the subjects spent about 90% of their time indoors. It seems that although ambient $PM_{2.5}$ measurements provide an adequate indicator of outdoor air quality for use in epidemiologic studies, such data may be inadequate to study the relationship between the health effects and non-ambient pollution generated indoors.

There was a significant but moderate correlation between log-transformed personal $PM_{2.5}$ concentration and personal B[a]P levels ($r = 0.43$). The correlation between ambient levels (indoors and outdoors) of $PM_{2.5}$ and the corresponding levels of B[a]P were in the same range. On average, only about 20% of variability in personal B[a]P could be explained by personal $PM_{2.5}$ level. This may be due to the fact that not all B[a]P is in the particulate form and that both

pollutants have somewhat different sources. Therefore, the extrapolation of personal exposure to B[a]P from personal $PM_{2.5}$ may lead to an underestimation of B[a]P exposure.

The study has confirmed a high correlation between B[a]P measured by monitors placed outside the apartments and the corresponding measurements indoors ($r = 0.95$). This strong correlation between outdoor and indoor measurements, especially in summer months, may derive from the fact that outdoor fine particles and B[a]P easily penetrate indoors and make a substantial contribution to indoor air quality. However, the high correlation between outdoor and indoor measurements in our study may also be partially explained by the fact that the outdoor monitors were placed in the direct neighbourhood of the apartment, at the window level.

A number of studies on personal $PM_{2.5}$ exposure [14–16, 20–23] reported exposure below the levels observed in our study. Several studies demonstrated also a relationship between personal, indoor and outdoor $PM_{2.5}$ in various residents and in various settings. For example, Rojas-Bracho et al. [14], in a study conducted in Boston, MA, on 18 individuals with COPD, found the median and mean personal exposure concentrations to be higher than the outdoor concentrations. Studies were carried out in Amsterdam and Helsinki in the winter and spring of 1998 and 1999. In the Amsterdam study, with 338–417 observations, the mean $PM_{2.5}$ concentrations were 24.3, 28.6, and 20.6 $\mu g/m^3$ for personal, indoor, and outdoor samples, respectively. When the measurements with ETS in the home were excluded, the mean indoor concentrations dropped to 16 $\mu g/m^3$, which was lower than outdoors. In the Helsinki study, the mean $PM_{2.5}$ concentrations were 10.8 $\mu g/m^3$ for personal, 11.0 indoor air and 12.6 $\mu g/m^3$ for outdoor air samples. Median Pearson's correlation coefficients between personal exposure and outdoor concentrations were 0.79 in Amsterdam and 0.76 in Helsinki. The median Pearson's correlation for the indoor/outdoor relationship was 0.85 for the Amsterdam study, excluding the homes with ETS. The correlation for indoor versus outdoor levels was 0.70 for all homes.

In the study by Pelizzari et al. [21], pooled correlations were derived for personal, indoor, and outdoor fixed-site ambient measurements in 732 participants aged 16 years or more. For PM_{25}, the mean concentrations were 28.4 $\mu g/m^3$ for personal exposure, 21.1 $\mu g/m^3$ for indoor samples, and 15.1 $\mu g/m^3$ for outdoor samples. A low but significant correlation ($r = 0.23$, $p \leq 0.01$) was reported between personal exposure and ambient exposure.

A number of studies used statistical modeling techniques to examine the relationship between ambient PM concentrations and personal exposure [23–25]. Data analysis involved the use of air exchange rates, penetration factors, and indoor/outdoor ratios, as well as examining exposure in various microenvironments (workplace, outdoors, indoors, exposure to smoke) and activities (traveling, cooking). The conclusion of the studies was that ambient aerosols contribute about 50% or more to personal PM exposure.

Previous reports on the urban air levels of B[a]P indicate that the annual mean ambient B[a]P concentrations measured in European cities in the 1950s and 1960s ranged from 1 to over 100 ng/m^3, the highest being found in the cities where coal was widely used for domestic heating. The policies for air pollution control and the changes in the fuels used since then have led to a reduction in the concentrations, and the available measurements indicate that annual ambient means in major urban areas in Europe are now within the range of 1–10 ng/m^3 B[a]P per m^3 [13]. Several studies in the United States and Europe have reported lower B[a]P levels than those observed in Kraków. Paired indoor and outdoor PAH measurements in 125 homes in Riverside, California [27], showed the median indoor air level of B[a]P of 0.19 ng/m^3 and the median outdoor level of 0.16 ng/m^3. B[a]P concentrations in 90 % of homes were below 0.65 ng/m^3, but in two homes the levels exceeded 5 ng/m^3. In another study on indoor and outdoor ambient air quality conducted in Pavia, Italy [28], B[a]P level in summer was lower than in winter, and indoor concentrations were many times as low as the levels outdoors. In eight homes, B[a]P levels indoors were lower than 0.21 (mean 0.11 ng/m^3), the outdoor levels ranging from 0.68–2.85 ng/m^3 (mean 1.19 ng/m^3).

In a study performed in Tokyo and Beijing during winter, both indoor and outdoor concentrations of B[a]P were

below 4 ng/m^3 in Tokyo. However, in Beijing, in a residential area where a mixture of coal, coal gas, and natural gas were used for heating, the indoor concentrations of benzo[a]pyrene were within the range of 1–40 ng/m^3 and the outdoor levels were slightly higher. In both cities, the indoor and outdoor B[a]P levels highly correlated with each other.

Summing up, we may conclude that although ambient $PM_{2.5}$ measurements provide an adequate indicator of outdoor air quality for use in epidemiologic studies, they may not provide adequate information for studies on the relationship between non-ambient pollution generated indoors and health effects. Our study has shown that B[a]P exposure of Kraków inhabitants would be in the range of many other urban areas worldwide. However, Kraków inhabitants experience in their life span an exposure to repeated spells of high concentrations of B[a]P during winter. The accumulation of repeated spells of exposure over winter periods may be implicated in adverse health effects like an excess risk of lung cancer, chronic non-specific obstructive disease, allergy, adverse birth outcomes, or even cardiovascular events. Since only about 20% of variability in personal B[a]P exposure could be explained by personal $PM_{2.5}$ level, the extrapolation of personal exposure to B[a]P from personal $PM_{2.5}$ data may be underestimated.

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